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Assessing the Impact of Oyster Reef and Living Shoreline Restoration on Macroinvertebrate Community Assemblages in Mosquito Lagoon, Florida

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ASSESSING THE IMPACTS OF OYSTER REEF AND LIVING SHORELINE RESTORATION ON MACROINVERTEBRATE COMMUNITY ASSEMBLAGES IN MOSQUITO LAGOON, FLORIDA

by

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ABSTRACT

As the world continues to experience substantial rates of habitat loss, habitat restoration has become of prime interest to ecologists worldwide. Restoration has shown to be successful in recovering targeted components of certain ecosystems but it is important to achieve a holistic understanding of the resulting ecological impacts it has on communities. To address this, four oyster reefs and three living shorelines were restored during the summer of 2017. These sites, along with four dead oyster reefs, four living oyster reefs, and three undisturbed (control) living shorelines, were sampled before restoration and regularly post-restoration for one year using lift nets. Macroinvertebrates were collected and enumerated in the lab. Diversity indices, community composition, and similarity percentages were then calculated and compared across treatments, time, and treatment-by-time. Live reefs displayed significantly higher species richness and Shannon diversity than restored and dead reefs. Simpson diversity did not differ between live and restored oyster reefs but both were significantly higher than dead reefs. Though not statistically detectable, species richness and Shannon diversity on restored reefs were relatively similar to dead reefs before restoration but became increasingly similar to live reefs over the course of the study. Additionally, analyses revealed significantly different community compositions between live reefs and restored reefs, as well as between live and dead reefs. Living shorelines showed no significant differences in diversity indices but did experience similar seasonal fluctuations in diversity across treatments. Just as with oyster reefs, restored and control living shorelines harbored significantly different communities across time. The findings of this study emphasize the need for dedication to thorough monitoring and multi-metric

evaluation of success in restoration efforts. This study and future research will equip resource managers with ways to quantify the effects of restoration.

DEDICATION

This thesis is dedicated, in loving memory, to Harry M. Searles Jr. and Kristin K. Goodnight. Thank you, for everything. *Soli Deo Gloria*.

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INTRODUCTION

Habitat Restoration

The biosphere has undergone significant change over the past several centuries via anthropogenic habitat conversion (Goldewijk, 2001; Gaston and Goldewijk, 2003; Hoekstra et al., 2005, Hanski, 2011). It is estimated more than half of land cover in certain biomes, such as temperate grasslands, temperate forests, and tropical and subtropical forests, have been lost to conversion in the past 300 years. As a result, we have witnessed significant reductions in biodiversity and ecosystem services in regions across the globe (Goldewijk, 2001; Brooks et al., 2002). Therefore, it has become established among ecologists that habitat loss is, currently, the greatest threat to global biodiversity (Brooks et al., 2002; Hanski, 2011). In response, countless scientists have adopted habitat restoration as the crux of their research efforts (Hobbs and Norton, 1996). Traditional habitat restoration involves identifying degraded habitats and modifying them as to revert them to their natural state. This is often accomplished by introducing native foundation species from which the ecosystem can develop naturally (Hobbs and Norton, 1996; Huxel and Hastings, 1999). By addressing spatially specific causes, habitat restoration can effectively reverse the negative effects of degradation, such as reductions in biodiversity and ecosystem productivity (Grimbacher at al., 2007; Van Katwijk et al., 2009).

While many restoration projects have been successful in terms of the survival of targeted foundation species, very few have maintained post-restoration monitoring to quantify the recovery of community-level processes occurring as a result of restoration (Bernhardt et al., 2007; Miller et al., 2010). Studies that have obtained adequate funding and the logistic means to do so have produced mixed results regarding the efficacy of foundation species survival to assess the holistic ecological effects of restoration (Luckenbach et al., 2005; Grimbacher et al., 2007;

Miller et al., 2010). In fact, there is often a lag between the recovery of foundation species and measurable community-level impacts. For example, Grimbacher et al. (2007) found that time since restoration had a large effect on beetle diversity in restored plots of tropical rainforest. Additionally, Luckenbach et al. (2005) observed that the occurrence of market-sized eastern oysters did not reliably predict the progress of many community-based metrics, including the abundance and diversity of several invertebrates and finfishes.

The apparent lack of understanding surrounding the community-level effects of restoration necessitates an in-depth investigation into changes in specific ecosystem components over an extended temporal scale. This study aims to elucidate these processes by monitoring changes in macroinvertebrate assemblages as a result of a large-scale intertidal oyster reef and living shoreline restoration program in Mosquito Lagoon, FL. This work will produce critical information regarding the temporal and spatial variation in community-level recovery post-restoration on estuarine oyster reefs and living shorelines. Such information will be vital to the improvement of restoration methods and post-restoration monitoring and management as anthropogenic pressures continue to degrade valuable marine habitat across the globe.

Intertidal Oyster Reefs

The eastern oyster (*Crassostrea virginica*; Gmelin, 1791) serves several critical roles in estuarine ecosystems. Firstly, eastern oysters remove excess nutrients and suspended particles from the water column via filter-feeding (Cole et al., 2015). This filtration strategy can effectively produce hundreds of gallons of clean water per day, the importance of which is augmented by increasing rates of eutrophication in estuaries worldwide. Intertidal oyster reefs also act as crucial ecosystem engineers in many estuaries by creating complex, three-dimensional structures that provide high quality habitat for countless sessile and motile marine organisms (Jones et al., 1994; Meyer, 1994; Tolley and Volety, 2005). This high-quality habitat supports unique communities of macroinvertebrates compared to adjacent habitats such as, soft-sediments and seagrass. Oyster reefs have also been found to sustain significantly higher abundances of species that are common to oyster reefs and other adjacent habitat types (Shervette and Gelwick, 2007; Gain et al., 2016).

In addition to providing physical habitat for many species, intertidal oyster reefs provide fertile hunting grounds for Xanthid crabs such as, stone crabs (*Menippe mercenaria*; Say, 1818) and mud crabs (e.g., *Panopeus herbstii*; Milne-Edwards, 1834 and *Eurypanopeus depressus*; Smith, 1869) and essential spawning grounds for several benthic fish species including, the naked goby (*Gobiosoma bosc*; Lacepède, 1800) and skilletfish (*Gobiesox strumosus*; Cope, 1870) (Menzel and Hopkins, 1956; Meyer, 1994; Breitburg, 1999). Due to their role as a foundation species, oyster reefs simultaneously provide critical foraging and spawning habitat as well as refugia from predators for a multitude of species that act as critical links in complex food chains of coastal ecosystems. Thus, it is expected that living oyster reefs should support a higher diversity of species than dead oyster reefs (i.e. mounds of disarticulated shell) as they provide higher quality habitat. Additionally, dead reefs that are restored should witness an increase in diversity as oyster densities increase over time after an initial crash in diversity immediately following restoration. Diversity will cease increasing at restored reefs once it becomes relatively similar to the species assemblage at live reefs. The same logic follows for community compositions on restored reefs. Dead reefs and living reefs will initially support unique communities but should become increasingly similar over time post-restoration.

Macroinvertebrates, specifically those occupying intertidal oyster reefs, are essential to the diet of many economically and ecologically important fish species (Plunket and La Peyre, 2005; Yeager and Layman, 2011). These crustaceans and other invertebrates also make up a large percentage of wading bird diet (Connolly and Colwell, 2005; Britto and Bugoni, 2015). Oyster reef-dwelling invertebrates therefore fill intermediate trophic levels; acting as predator and prey, these invertebrates create critical connections between higher and lower trophic level species, making them an integral part of estuarine food webs. Because of the important roles these invertebrates fill, many studies have been published quantifying the diversity and abundance of macroinvertebrates on oyster reefs, some of them in ML (Boudreaux et al., 2006; Barber et al., 2010). Though studies quantifying the effects of oyster reef restoration on invertebrate and fish community diversity have been conducted elsewhere (Luckenbach et al., 2005; Geraldi et al., 2009), to my knowledge none utilizing a BACI design have been attempted in the IRL.

Living Shorelines

Several shell midden sites within ML (e.g. Turtle Mound) have undergone significant erosion due to rising sea levels and other extreme climatic events such as hurricanes (Hellmann, 2013; Donnelly et al., 2015). These ancient shell middens are historically significant as Timucuan and Ais Native Americans constructed them over the past 2000 years (Hellmann, 2013; Donnelly et al., 2015). To preserve these archaeological sites, Canaveral National Seashore (CANA) and the Coastal and Estuarine Ecology (CEE) Lab at UCF began collaborating in 2009 to conduct shoreline restoration and stabilization (Donnelly et al., 2015).

Such restoration efforts have been accomplished by implementing wave dispersion materials (i.e. oyster shell or boulders) and native plants in the intertidal zone to prevent erosion (Crooks and Turner, 1999; Piazza et al., 2005; Donnelly et al., 2015). Living shoreline restorations have been extremely successful in several coastal ecosystems such as, Dauphin Island, Alabama; Outerbanks, North Carolina; and the Chesapeake Bay (Davis et al., 2006; Swann, 2008; Gittman et al., 2016). Living shorelines have been shown to reduce erosion and disperse wave energy with great efficiency. The associated marsh grasses and trees have also been shown to sequester excess nutrients from their surroundings, a key service in coastal ecosystems under heavy anthropogenic stress (COPRI, 2014). In addition, living shorelines support significantly greater species richness and diversity than man-made stabilizers, such as bulkheads and seawalls, due to increased habitat availability (Seitz et al., 2006; Lucrenzi et al., 2010). Because of the numerous benefits of living shorelines there has been an increasing interest in restoration of these ecologically unique sites in the IRL.

The effects of restoration on community structure around living shorelines are of particular ecological interest in ML due to the occurrence of seagrass beds just below the intertidal zone. Seagrass beds are known to provide high quality habitat and foster high biodiversity across the globe (Short et al., 2007; Dorenbosch et al.). The IRL is no exception as seagrass beds can harbor hundreds more species and individuals than adjacent sandy bottom habitats (Gilmore, 1995). However, seagrass beds are particularly vulnerable to water clarity, erosion, and subsequent burial via anthropogenic changes to sedimentation dynamics (Marba and Duerte, 1994; Cabaco et al., 2008; Ruiz and Romero, 2008). Shoreline stabilization is implemented with the goal of preventing erosion and thus burial of seagrasses. However, the act of restoration causes significant disturbance, which should initially act opposite to the diversitybolstering effects of seagrass. Thus, decreases in diversity shortly after restoration are expected. Over time there should be an increase in diversity at restored sites. This increase is expected to stop when the diversity of restored sites is no longer relatively similar to non-eroded shorelines. Similar patterns should exist for species composition as well. Degraded sites should support unique communities from restored sites, which will begin to resemble one another over time. This study aims to test these hypotheses to obtain a thorough understanding of post-disturbance recovery in the marine environment for the improvement of future restoration efforts and preservation of biodiversity.

METHODS

Study Area

Mosquito Lagoon (ML; 28.835940˚ N, 80.796794˚ W) is the northernmost basin of the broader Indian River Lagoon (IRL) and is connected to its remainder via Haulover Canal, a man-made channel located in the southern half of the estuary. Ponce De Leon Inlet defines the northern end of ML and is the estuary's only ocean access (Fig 1). Due to its location in a biogeographic transition zone, the IRL is one of the most diverse estuaries in North America and is home to over 400 fish species (of the 782 species found on the entirety of the east coast of central Florida) as well as several hundred species of invertebrates, birds, and marine mammals (Gilmore 1995; Swain, 1995; Tremain and Adams, 1995; Paperno et al., 2001; Smithsonian Institution 2006).

Oyster Reef and Living Shoreline Restoration

Figure 1. Map of Mosquito Lagoon.

The methods replicated in this study for oyster reef restoration have been previously successful in ML (Garvis et al., 2015). Dead reef sites, which are composed of disarticulated oyster shell stacked over a meter above the water's surface, were leveled to the intertidal zone by four to six-person teams using shovels and pickaxes. Disarticulated oyster shells were attached to aquaculture-grade mesh mats (36 shells per 0.25 m^2 of mat) and placed on the leveled reef area. Each corner of the oyster mats was then anchored to donut weights. This procedure anchored the developing reef, making it resistant to uprooting and toppling by natural wave action and anthropogenic boat wakes (Garvis et al., 2015).

The techniques used to restore the four impacted living shoreline sites were modified from Donnelly et al. (2017). Red mangroves (*Rhizophora mangle)* were planted along the upper intertidal zone (2 plants/m) followed by *Spartina alterniflora* in the mid-intertidal (3 plants/m) to ensure sediment stabilization and wave energy attenuation. The site was then lined with two-gallon bags of dead shell in the lower intertidal to maximize wave energy attenuation.

Sampling Design

A Before-After-Control-Impact (BACI) design was implemented on twelve oyster reefs starting in May 2017. Four reefs were designated for restoration, four dead reefs were designated as negative controls, and four live reefs were selected as positive controls (Fig. 2). Seven living shoreline sites were selected for this study, four restoration sites and three control sites (Fig. 3). Due to the impacts of Hurricane Irma the southernmost living shoreline site was lost to severe erosion and has been dropped from our analyses. Otherwise, each site was sampled at each of the following time periods: 1 week prior to restoration, 1 week following restoration, 2 weeks after restoration, 4 weeks after restoration, 28 weeks after restoration, 40 weeks after restoration, and 48 weeks after restoration.

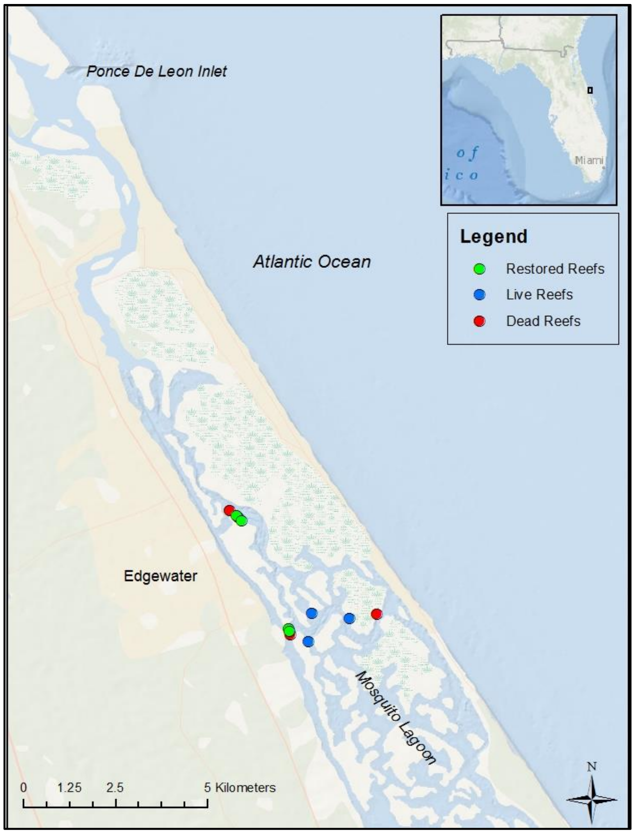

Figure 2. Oyster reef sampling sites.

Figure 3. Living shoreline sampling sites.

These reefs and shorelines were sampled for macroinvertebrates using lift nets. Lift nets were originally designed by Crabtree and Dean (1982) and were modified for use in Florida by Coen et al. (1996). They were constructed using PVC and 2mm mesh netting. The PVC was made into 0.5m x 0.5m square frames and a 0.5m deep mesh net was attached using Zip Ties. Six nets were deployed on each oyster reef for a total of seventy-two lift nets. On oyster reefs, three lift nets were placed in the low intertidal zone and three were placed in the high intertidal zone. At living shoreline sites six nets were placed equidistant along the length of the 75m-long site in the mid intertidal zone. At oyster sites each net held a mesh mat with several disarticulated oyster shells attached to it. These oyster mats simulate habitat cover for fish and invertebrates and have been found to be highly effective at sampling invertebrates (Barber et al. 2006). Two and a half gallons mesh bags filled with disarticulated oyster shells were used for the same effect at living shorelines as they more accurately mimicked the corresponding habitat. All nets were deployed 1 week prior to sampling. Upon collection, researchers retrieved nets by slowly approaching them and pulling up swiftly to trap any fauna utilizing the disarticulated oyster shells. Upon collection of lift nets, their accompanying oyster mats and shell bags were left in place in the water near the sampling sites. All invertebrates captured were stored in 70% ethanol and brought back to the lab for enumeration, measuring, and weighing.

Data Analyses

In order to answer my proposed research questions, two-way ANOVAs were conducted to test the effect of treatment and sampling time on each measure of diversity. Species richness values were used as a basic indicator of species carrying capacity of each treatment (i.e. reef type). Species richness is equal to the number of unique species found in a given sample.

The Shannon-Wiener diversity index was used to compare community diversity of each treatment over time. The value of the Shannon index provides a measure of the abundance and evenness of the species in a given area of interest and relies more heavily on the richness of the community than other metrics (Nagendra, 2002). Shannon diversity is calculated as the negative sum of the proportion of each species multiplied by the natural logarithm of that proportion (Equation 1; Shannon and Wiener, 1963; Nagendra, 2002), where p_i is the proportion of a given species, S is the number of species, and H is the Shannon diversity value.

$$H = -\sum_{i=1}^{S} (p_i * ln(p_i)) \qquad \textbf{Equation 1}$$

Simpson's diversity index was also be measured and used to compare biodiversity across treatments and time. Simpson's diversity is a metric used for comparing species evenness as well as diversity. It accounts for high variability in abundances between species and therefore provides a less biased estimate of diversity in relatively uneven communities. Simpson diversity is calculated as the following where n is the number of individuals of a given species, N is the number of total individuals, and D is Simpson diversity (Equation 2; Simpson, 1949; Nagendra, 2002):

$$D = 1 - \left(\frac{\Sigma\, n(n-1)}{N(N-1)} \right) \qquad \textbf{Equation 2}$$

Pielou's evenness was also considered apart from Simpson's diversity. Pielou's evenness is a comparison between the relative abundances of the species in a community. A higher value indicates more equal abundances among species (Rueda and Defeo, 2003). Pielou's Evenness is calculated by dividing the Shannon diversity value of a community by the natural logarithm of species richness, where J is Pielou's evenness, H' is Shannon diversity, and s is species richness (Equation 3).

$$\mathrm{J} = \frac{H'}{\ln(\mathrm{s})} \qquad \textbf{Equation 3}$$

In order to ensure a comprehensive analysis of the effects of restoration, I also examined changes in community composition post-disturbance (i.e. restoration). To formally test my hypotheses regarding the differences in community composition across time and treatments, pairwise analyses of similarities (ANOSIM) were conducted on each treatment-time. ANOSIM is a non-parametric hypothesis test that relies on the ranks of similarities between samples within a resemblance matrix (Clarke, 1993). ANOSIMs yield a *p*-value that can be used to test the null hypothesis that no significant dissimilarity exists between sites. SIMPER (similarity percentage) analyses were also conducted to determine percent contribution of each species across treatments and time. SIMPER analyses reveal pairwise percent dissimilarity values for each treatment over time. These data complemented the ANOSIMs by yielding a quantifiable metric for determining what species are driving any observed differences. These combined analytical methods allow the creation of a comprehensive view of the dynamics of macroinvertebrate communities post-restoration.

RESULTS

Oyster Reef Diversity

In total, 6,385 macroinvertebrates belonging to 35 species were collected from oyster reefs between May, 2017 and June, 2018. A full enumeration of these species is detailed in Table 1. Live reefs yielded 2,221 individuals, the most of any treatment. Live reefs were followed by restored reefs, from which 2,148 invertebrates were collected. Dead oyster reefs produced the fewest invertebrates at 2,016. Live reefs also had significantly higher species richness than both restored and dead reefs (Two-way ANOVA $F_2 = 6.11$, $p = 0.003$, posthoc Tukey HSD p adj. = 0.0367, 0.0038). Dead and restored reefs did not display significantly different species richness. Though significant differences existed between treatments, there were no significant differences between treatments within any sampling time. There was, however, an increasing trend in species richness at restored reefs post-restoration (Fig. 4). Live reefs had significantly greater Shannon diversity than both dead and restored oyster reefs throughout our sampling time frame (Two-way ANOVA $F_2 = 5.85$, $p = 0.004$, posthoc Tukey HSD p adj. = 0.0067, 0.0232). Just as there was an increasing trend in species richness post-restoration, there was an upward trend in Shannon diversity on restored reefs after the 1-week sampling period (Fig. 5). Simpson diversity was shown to be significantly greater at live reefs and restored reefs than at dead reefs (Two-way ANOVA $F_2 = 5.39$, $p = 0.007$, posthoc Tukey HSD p adj. = 0.009, 0.037). Additionally, there was no significant difference in Simpson diversity between live reefs and restored reefs. Simpson diversity also dropped one week after restoration and then began to rise over time until the 40-week sampling period (Fig. 6). Lastly, there was no significant difference in Pielou's Evenness between all treatments. No obvious patterns revealed themselves on restored reefs post-restoration, save the decrease in evenness at the 40-week sampling period across all treatments. However, all treatments did appear to follow similar temporal fluctuations (Fig. 7).

Table 1. Counts by species at oyster reefs.

Species	Count
Alpheus heterochaelis	162
Alpheus spp	19
Amphiodia pulchella	1
Amphipholis squamata	33
Amphiura spp	1
Callinectes ornatus	22
Callinectes sapidus	19
Callinectes similis	7
Callinectes spp	2
Charybdis hellerii	3
Clibinarius vittatus	61
Eurypanopeus depressus	288
Eurypanopeus dissimilis	1
Eurytium limosum	1
Farfantepenaeus aztecus	3
Farfantepenaeus brasiliensis	1
Farfantepenaeus duorarum	4
Hippolyte spp	29
Isognomon alatus	1
Leander spp	11
Libinia dubia	5
Litopenaeus setiferus	1
Macrobranchium spp	62
Menippe mercenaria	7
Neopanope sayi	68
Palaemon spp	16
Palaemonetes spp	51
Palaemonetes spp	1
Panopeus herbstii	386
Panopeus simpsoni	175
Periclimenaeus spp	170
Petrolisthes armatus	4315
Petrolisthes galathinus	33
Rhithropanopeus harrisii	424
Synalpheus spp	2
Total	6385

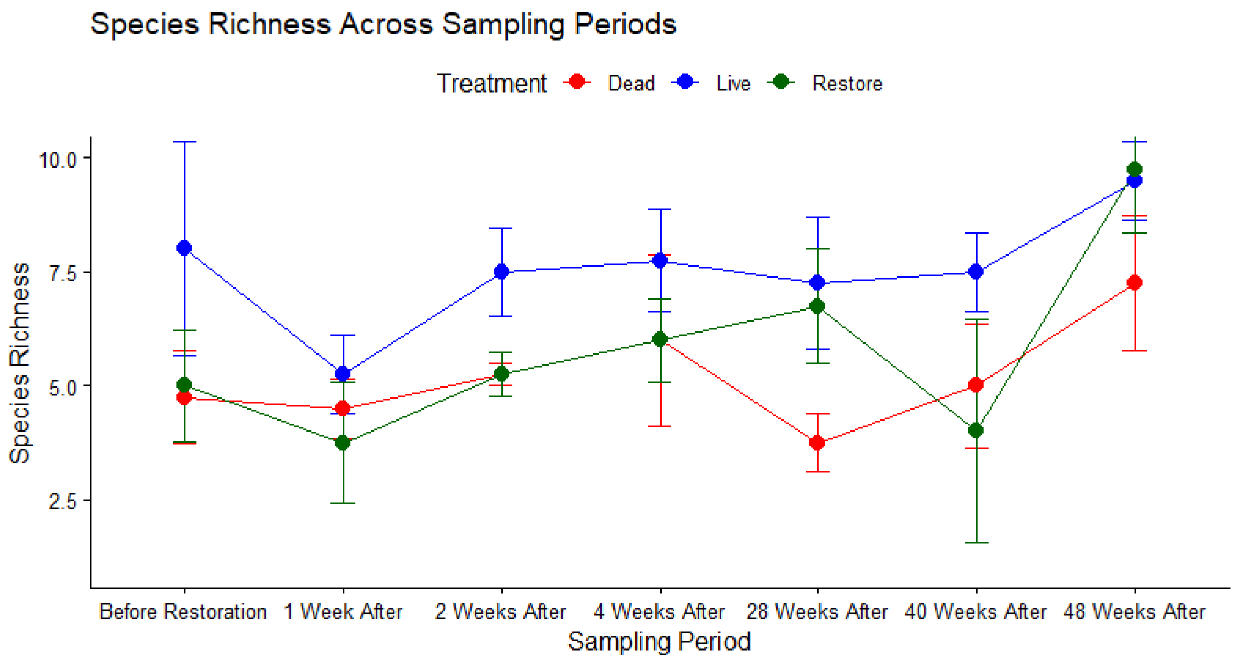

Figure 2. Species richness values by oyster reef treatment over time.

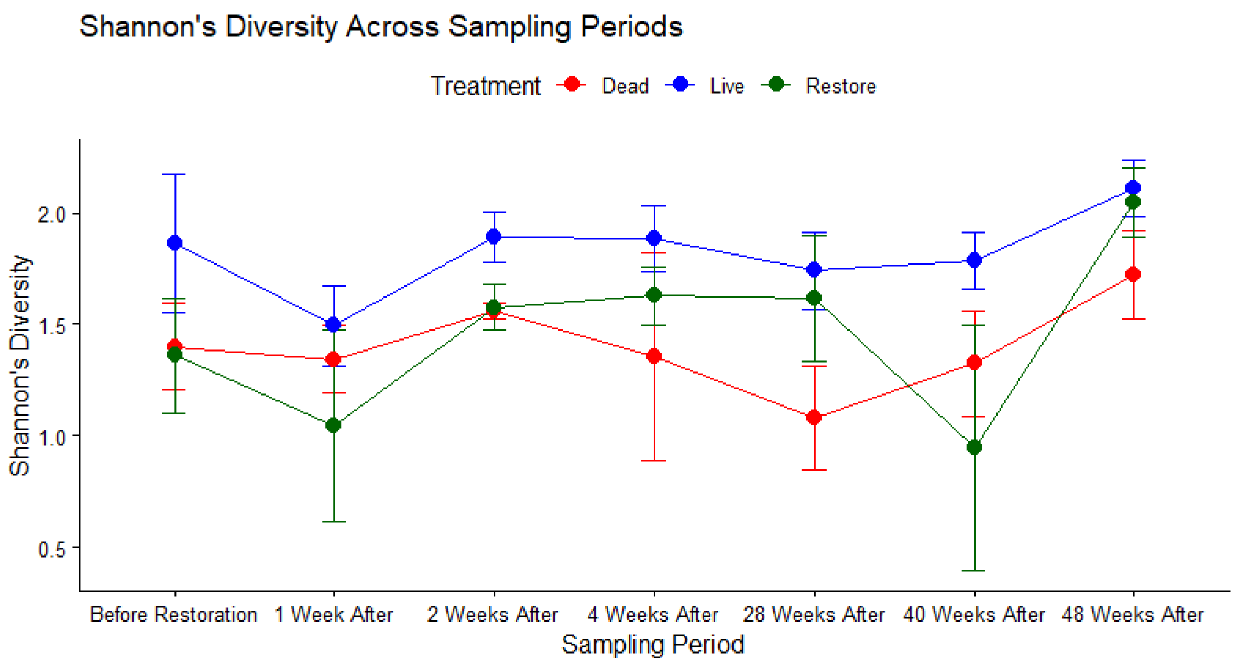

Figure 3. Shannon Diversity by oyster reef treatment over time.

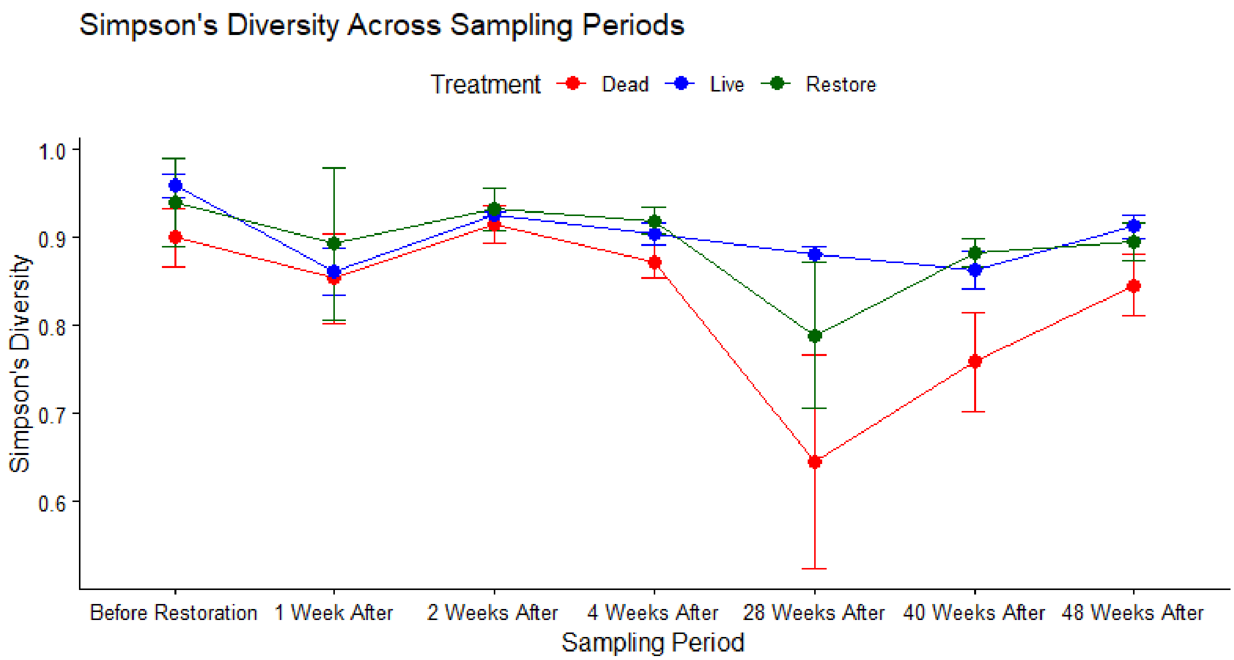

Figure 4. Simpson Diversity by oyster reef treatment over time.

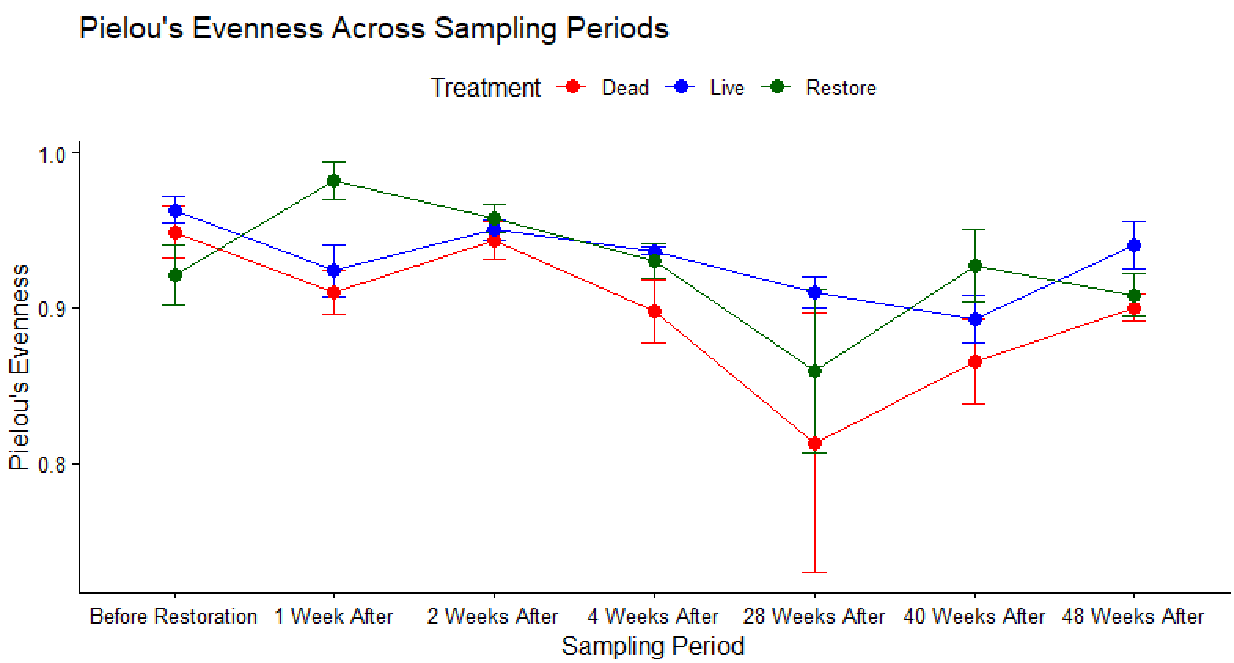

Figure 5. Pielou's Evenness by oyster reef treatment over time.

Oyster Reef Community Composition

Analyses on oyster reef dataset revealed significant dissimilarities between oyster reef macroinvertebrate communities across time, treatments, and treatments-by-time (ANOSIM p = 0.001, 0.002, 0.001, respectively). Subsequent examination of sampling times, with all treatments pooled, revealed that the before restoration 1-week post restoration, 2-weeks post-restoration, and 4-weeks post-restoration times were not significantly different from one another. However, each of these early time periods were significantly different than later sampling times (i.e. 28 weeks, 40 weeks, and 48 weeks post-restoration). The only exception to this pattern was the similarity between the 4-week post-restoration and the 40-weeks post-restoration sampling periods, which were not significantly different. Additionally, the 40-weeks post-restoration and 48-weeks post-restoration sampling periods showed significantly different species compositions. There were significantly different species compositions between live and restored reefs (pairwise ANOSIM p = 0.035), and between live and dead reefs (p = 0.017). However, there was no statistically significant difference between restored and dead reef communities.

When community composition was assessed on treatment-times, results were ambiguous but, a few patterns emerged. The species composition of dead reefs before restoration did not significantly differ from that of restored reefs before restoration (pairwise ANOSIM p = 0.633). Additionally, live reefs before restoration were not significantly different than dead or restored reefs before restoration. Likewise, live reefs and dead reefs one week after restoration as well as restored and dead reefs one week after restoration showed no statistically significant differences in species composition. However, restored reefs one week after restoration displayed significant differences in community composition than live reefs (p = 0.022). No significant differences were observed across treatments at the 2-week sampling period. Additionally, no significant differences were observed between live reefs and dead reefs or between dead reefs and restored reefs at the 4-week sampling period. However, there was a significant difference between live reef and restored reef community composition at the 4-week sampling period (ANOSIM p = 0.027).

Due to Hurricane Irma's impact on Florida, field work was impossible during the 12-week sampling period. The next collection period was 28 weeks after restoration. No significant differences in species composition were observed between live reefs and dead reefs, between restored and dead reefs, or live and restored reefs at this sampling period. Conversely, live reefs and dead reefs did display significantly different species compositions at the 40-weeks post-restoration sampling period (p = 0.006). Live reefs and restored reefs 40 weeks post-restoration also supported significantly different species compositions (p = 0.003). Dead reefs and restored reefs did not display significantly different species compositions 40 weeks post-restoration (p = 0.574). Finally, all treatment types did not display significantly different species compositions 48 weeks post-restoration (dead-live p = 0.116, dead-restored p = 0.128, live-restored p = 0.382).

There were high average dissimilarities between reef types. Live and restored reefs had an average dissimilarity of 86.68% with *Petrolisthes armatus*, *Panopeus herbstii*, *Eurypanopeus depressus*, *Rhithropanopeus harrisii*, *Alpheus heterochaelis*, and *Panopeus simpsoni* contributing to 80% of the dissimilarity between them. Live and dead reefs had an average dissimilarity of 87.08% with *Petrolisthes armatus*, *Panopeus herbstii*, *Eurypanopeus depressus*, *Rhithropanopeus harrisii*, *Alpheus heterochaelis, and Alpheus heterochaelis* contributing to 80% of the dissimilarity between them. Restored and dead reefs displayed an average similarity of 88.25% with *Petrolisthes armatus, Panopeus herbstii*, *Rhithropanopeus harrisii*, *Eurypanopeus* *depressus*, *Panopeus simpsoni*, and *Clibinarius vittatus* contributing to 80% of the dissimilarity between them.

Similarities within treatments were generally driven by very few species. For example, the average similarity within the live reef treatment was 16.83% with *Petrolisthes armatus*, *Eurypanopaeus depressus*, and *Panopeus herbstii* accounting for over 80% of the similarity. Restored reefs showed similar patterns with an average similarity of 12.5% with *Petrolisthes armatus* and *Panopeus herbstii* accounting for 80% of the similarity within the treatment. Dead reefs showed a 11.27% similarity with *Petrolisthes armatus* and *Panopeus herbstii* contributing to 80% of the similarity within the treatment (Table 2). Similarities within treatment-times were also driven by only a few species. The full results of the SIMPER analysis on treatment-times are detailed in Tables 3, 4, and 5 (Appendix).

Table 2. Average abundances and contributions of species by oyster reef treatment.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Live					
Petrolisthes armatus	0.97	10.88	0.61	64.66	64.66
Eurypanopeus depressus	0.32	2.36	0.36	14.03	78.69
Panopeus herbstii	0.26	1.23	0.25	7.3	85.99
Rhithropanopeus harrisii	0.23	0.8	0.22	4.78	90.77
Restored					
Petrolisthes armatus	0.77	9.19	0.5	73.54	73.54
Panopeus herbstii	0.18	1	0.23	7.99	81.53
Rhithropanopeus harrisii	0.21	0.75	0.19	6.04	87.57
Eurypanopeus depressus	0.13	0.62	0.18	4.94	92.51
Dead					
Petrolisthes armatus	0.8	8.01	0.44	71.08	71.08
Panopeus herbstii	0.21	1.17	0.21	10.41	81.49
Panopeus simpsoni	0.15	0.88	0.22	7.78	89.28
Eurypanopeus depressus	0.13	0.59	0.2	5.21	94.49

Living Shoreline Diversity

Over the course of the study. 4,801 macroinvertebrates belonging to 28 species were collected from living shorelines in the southern portion of ML. A full enumeration of these species is detailed in Table 6. Control sites produced 2,314 macroinvertebrates while Restored living shorelines yielded 2,487. Species richness did not significantly differ between treatments over the course of the sampling period. However, there was a significant difference in species richness between sampling times due to a significant decrease in species richness between the 4-week and 40-week restoration period (Two-way ANOVA $F_6 = 2.67$ $p = 0.036$, posthoc $p = 0.024$).

Shannon diversity showed no significant differences across treatments or times at living shoreline sites. Simpson diversity and Pielou's Evenness also showed no significant difference between control and restored sites ($p = 0.365$, 0.475). However, there was a significant decrease in Simpson diversity over time at living shoreline sites (Two-way ANOVA $F_6 = 5.93$ $p = 0.001$). The 40-week time period had significantly lower diversity than the before restoration, 1-week post-restoration, 2-weeks post-restoration, 4-weeks post-restoration, and the 48-weeks post-restoration periods (posthoc Tukey HSD $p = 0.005$, 0.003, 0.024, 0.0001, 0.021). All diversity indices displayed similar temporal trends with diversity reaching a minimum 40 weeks after restoration and peaking during the 4-week and 48-week sampling times (Fig. 8, 9, 10, 11).

Table 3. Counts by species at living shorelines.

Species	Count
Alpheus heterochaelis	273
Callinectes exasperatus	1
Callinectes ornatus	38
Callinectes sapidus	65
Callinectes similis	18
Clibinarius tricolor	8
Clibinarius vittatus	9
Eurypanopeus depressus	521
Farfantepenaeus aztecus	1
Hippolyte spp	10
Leander spp	565
Libinia dubia	27
Litopenaeus setiferus	8
Macrobranchium spp	1
Neopanopeus sayi	40
Palaemon spp	166
Palaemonetes spp	280
Panopeus herbstii	7
Panopeus simpsoni	7
Periclimenaeus spp	155
Petrolisthes armatus	2200
Petrolisthes galathinus	4
Polychaete spp	2
Rhithropanopeus harrisii	337
Sesarma curacaoense	1
Solenoceridae spp	1
Tozeuma carolinense	3
Uca spp	53
Total	4801

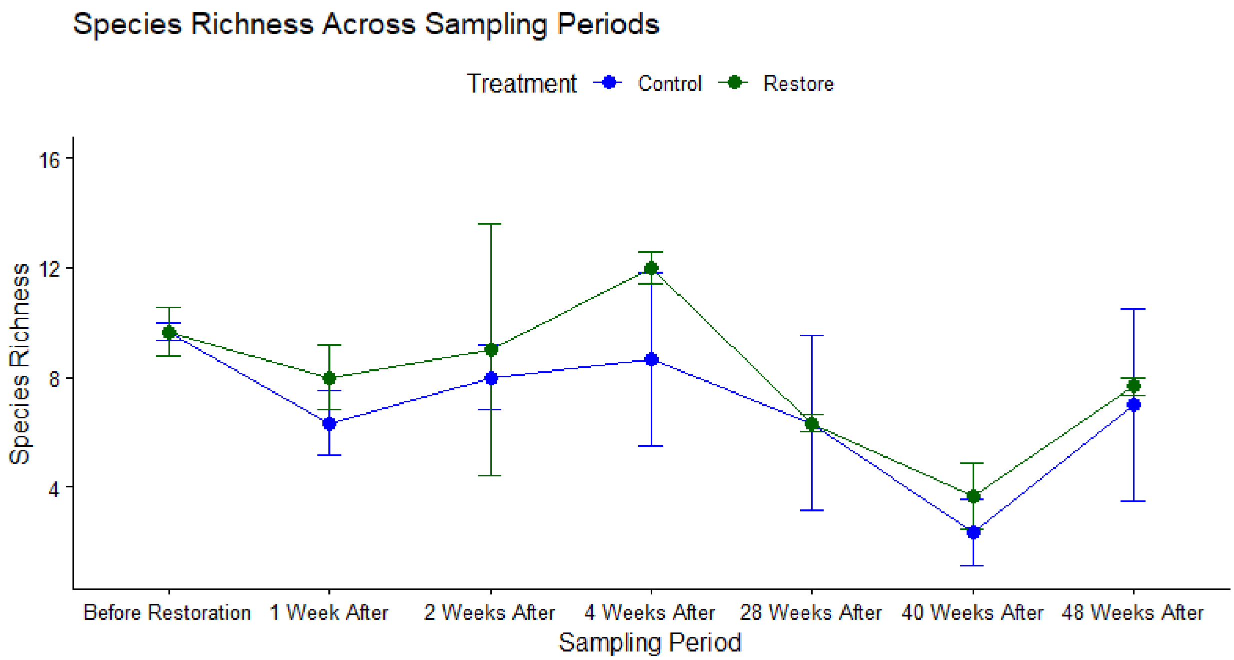

Figure 6. Species richness by living shoreline treatments over time.

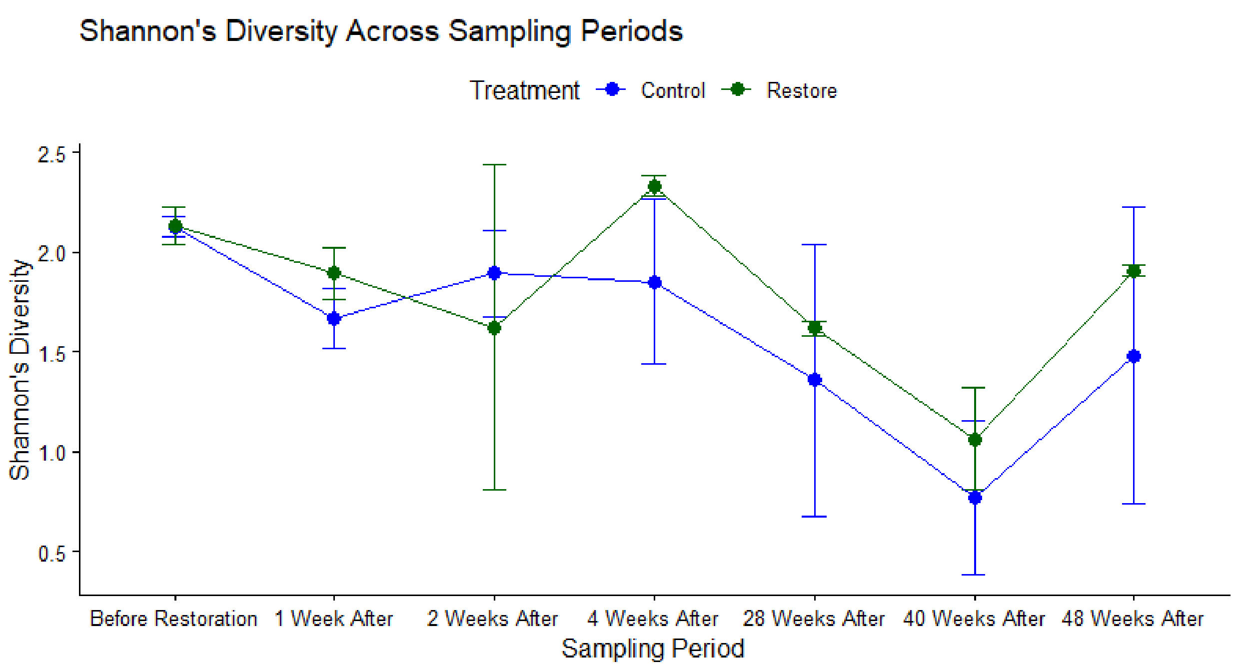

Figure 7. Shannon Diversity by living shoreline treatment over time.

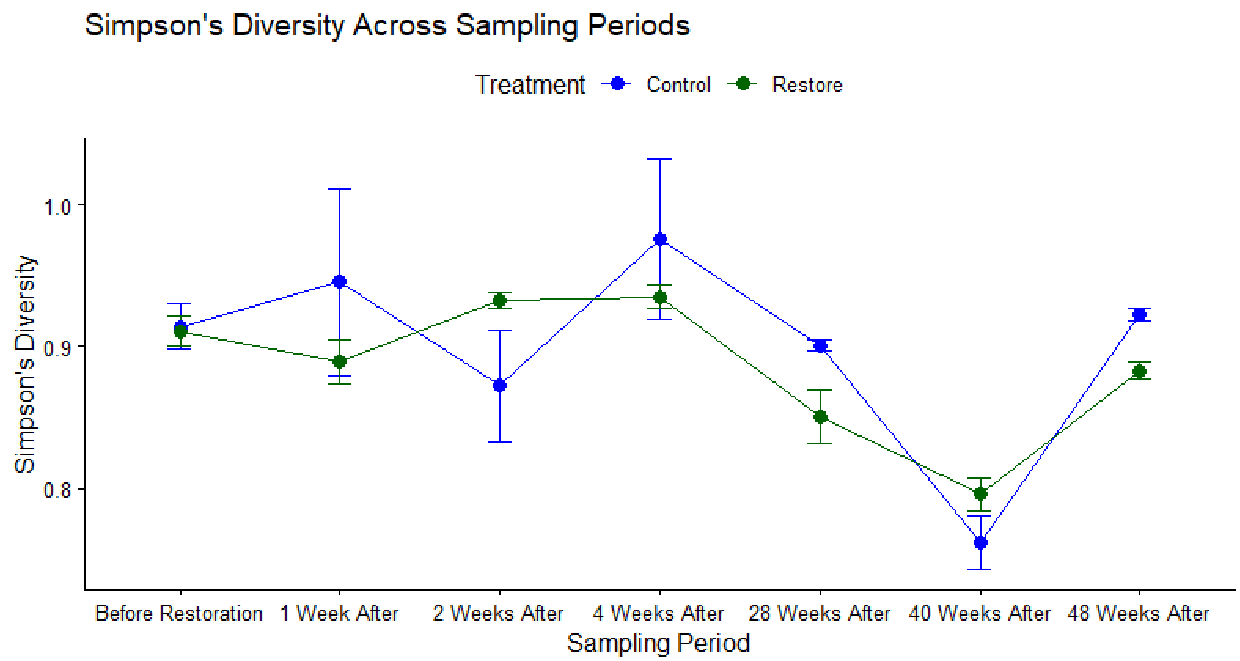

Figure 8. Simpson Diversity at living shoreline treatment over time.

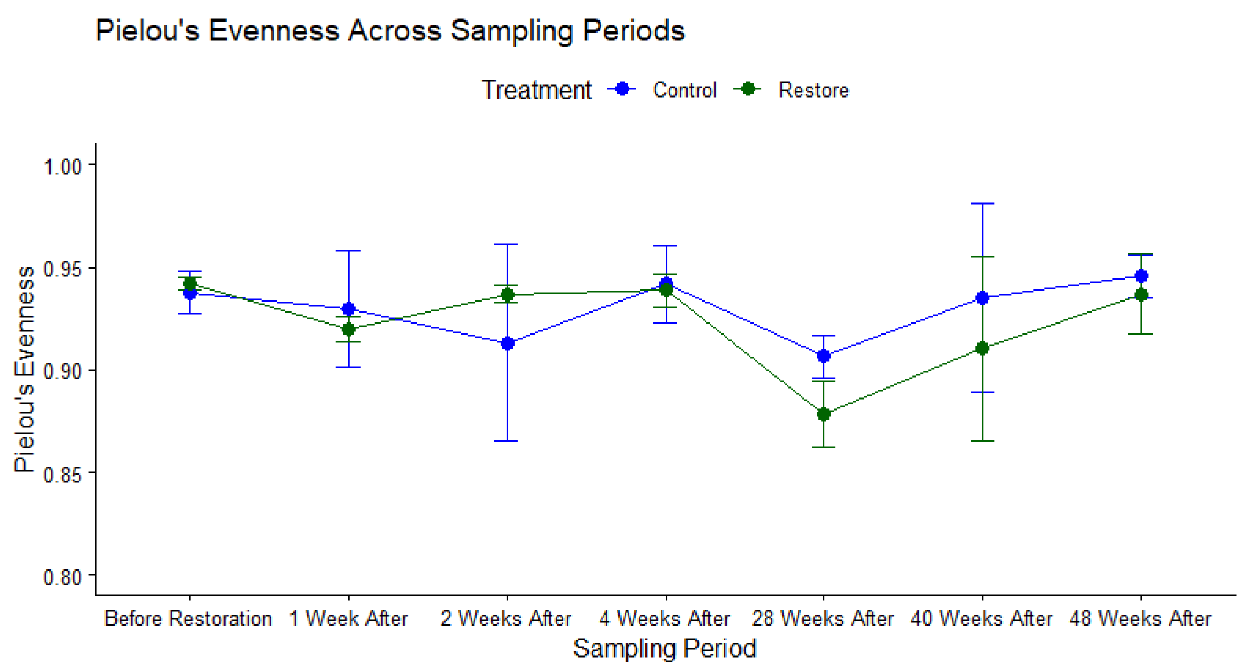

Figure 9. Pielou's Evenness by living shoreline treatment over time.

Living Shoreline Community Composition

Much like oyster reefs, the community composition at living shorelines were complex. However, noticeable patterns did emerge. There were significant differences in species composition were present between treatments, sampling times, and treatment-times (ANOSIM $p = 0.001, 0.003, 0.001$, respectively). Species composition was significantly different between all time periods, save one week and 28 weeks post-restoration, two weeks and four weeks post-restoration, as well as 28 weeks and 40 weeks post-restoration (pairwise ANOSIM $p = 0.227, 0.067, 0.18$).

There were no significant differences between restored and control living shorelines before restoration occurred. However, these treatments did display significant differences in species composition 1 week after restoration took place (pairwise ANOSIM $p = 0.007$). Species compositions then became more similar between restored and control sites during the 2-week sampling period as there was no significant difference in species composition. The 4-weeks post-restoration sampling period displayed significantly different species compositions once again, however (pairwise ANOSIM $p = 0.03$). Over the next several months, the two communities became increasingly similar as no significant difference was detected between restored and control sites. No significant differences were detected for the remainder of the sampling schedule with species compositions remaining relatively similar between restored and control living shorelines at 40 weeks and 48 weeks post-restoration.

Data revealed relatively high dissimilarity between restored and control living shorelines (average dissimilarity of 77.44%). Similarities within each treatment were, generally, driven by a low number of species (Table 7). For example, control sites had an average similarity of 20.15% with 90% of it being accounted for by *Petrolisthes armatus, Alpheus heterochaelis*, *Eurypanopeus depressus*, *Rhithropanopeus harrisii*, *Leander spp*, and *Palaemonetes spp*. *Petrolisthes armatus*, *Eurypanopeus depressus*, *Rhithropanopeus harrisii*, *Alpheus heterochaelis*, *Palaemon spp*, and *Leander spp* accounted for 90% of the similarity within the restored treatment, which had an average similarity of 29.47%. Similarities within treatment-times were also driven by few species. The similarities of each treatment-time are listed in Tables 8 and 9 (Appendix).

Table 4. Average abundances and contributions of species by living shoreline treatment.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Control					
Petrolisthes armatus	1.08	6.45	0.57	32.01	32.01
Alpheus heterochaelis	0.54	3.9	0.53	19.35	51.36
Eurypanopeus depressus	0.58	3.79	0.37	18.82	70.18
Rhithropanopeus harrisii	0.47	2.65	0.44	13.14	83.32
Leander spp	0.32	0.77	0.16	3.84	87.16
Palaemonetes spp	0.28	0.76	0.17	3.76	90.92
Restored					
Petrolisthes armatus	1.42	13.4	0.83	45.46	45.46
Eurypanopeus depressus	0.78	7.29	0.57	24.75	70.21
Rhithropanopeus harrisii	0.45	1.96	0.34	6.64	76.85
Alpheus heterochaelis	0.34	1.95	0.38	6.6	83.45
Palaemon spp	0.29	1.34	0.25	4.54	88
Leander spp	0.38	1.14	0.21	3.88	91.88

Oyster Reef and Living Shoreline Comparisons

Species richness not only varied within oyster reef and living shoreline habitats, but among them as well. Data indicate significantly higher species richness at restored living shoreline sites than at dead oyster reefs, live oyster reefs, and restored oyster reefs (Two-way ANOVA $F_4 = 14.22$, $p = 0.000001, 0.0002, 0.0002$, respectively). However, species richness was not significantly different between control living shoreline sites and all oyster reef treatments. Following the same pattern, Shannon diversity was significantly higher at restored living shorelines than at dead oyster reefs, and restored oyster reefs (p adj. < 0.0001, p adj. = 0.001). Unlike species richness, however, Shannon diversity displayed no significant difference between live oyster reefs and restored living shorelines. There was also no significant difference between living shoreline control sites and all oyster reef treatments. This was in contrast to living shoreline control sites which had significantly higher Simpson diversity than dead oyster reefs (Two-way ANOVA $F_4 = 5.49$ $p = 0.001$, posthoc Tukey HSD p adj. = 0.019). Lastly, Pielou's Evenness showed no significant differences between treatments.

Comparison of community composition at living shoreline and oyster reef treatments revealed significant differences in community composition within the dataset (ANOSIM p = 0.001). Control living shoreline sites harbored significantly different communities than dead oyster reefs, restored oyster reefs, and live oyster reefs (p = 0.001, 0.001, 0.002). Restored living shorelines also produced significantly different communities than dead, restored, and live oyster reefs (p = 0.001, 0.001, 0.003). Accordingly, there was SIMPER analyses yielded high average dissimilarities between the treatments, with control living shoreline sites having dissimilarities of 91.78%, 91.52%, and 87.51% with dead, restore, and live oyster reefs, respectively. Restored living shoreline sites also displayed high average dissimilarities of 90.01%, 89.92%, and 84.68% with dead, restored, and live oyster reefs.

DISCUSSION

Though the results of the study were generally more complex and not as clear cut as predicted, many expected trends in the data were indeed observed. Species richness were higher at living oyster reefs than at dead or restored reefs. Shannon diversity and Simpson diversity were also significantly higher at live reefs than dead reefs. These findings corroborate the conclusions of Shervette and Gelwick (2007) and Gain et al. (2016) that oyster reefs support a higher diversity and abundance of species than lower-quality habitats. In light of the agreement between results, I expected that there would be significant differences in each diversity index between live and dead reefs at each treatment-time. However, no such significance was observed. As each time period can be viewed as a subset of samples within the entire temporal sampling domain, the sample size within each treatment-time may have been too small to detect significant differences in diversity. Even so, inspecting the data graphically can yield useful insight into changes in communities post-restoration.

Diversity values were close at restored and dead oyster reefs before restoration (Figs. 4-7). Diversity then dropped 1 week after restoration at restored sites. From there, species richness and Shannon diversity increased over time until the 40-weeks sampling period. There, restored treatments experienced large decreases in both indices. Simpson diversity and Pielou's Evenness experienced similar drops at the 28-week time period. After their respective drops, each index increased to previous levels. Species richness and Shannon diversity even increased to higher levels than previously seen at restored reefs and were almost equal with live reefs 48 weeks post-restoration. These results indicate that new species colonized restored oyster reefs over the course of the study period. Some even appeared as little as two weeks post-restoration. Though not statistically detectable, increases in diversity did occur at restoration sites over time. Thus, the inclusion of data collected from a second year of sampling may increase the sample size adequately to enable detection of statistical significance, and suggests the need for protracted monitoring. This echoes the argument presented by Luckenbach et al. (2005) that long-term ecological monitoring is necessary to adequately quantify the success of restoration.

Unlike oyster reefs, living shorelines did not display significantly different diversities between treatments. Once again, examining these data graphically yielded valuable information regarding post-restoration processes (Figs. 8-11). Species richness, Shannon diversity, and Simpson diversity were all nearly identical between restored and control living shorelines before restoration. As predicted, there was a drop in all of these indices one week after restoration. Within four weeks of restoration species richness and Shannon diversity were higher at restored sites than control sites. Additionally, Pielou's Evenness and Simpson diversity fluctuated regularly with neither treatment maintaining higher values for an extended period of time. These results indicate that new species colonized restored living shorelines within two weeks of the restoration event. Additionally, relatively similar species evenness values (i.e. Simpson diversity and Pielou's evenness) over time suggest these new species colonized successfully. If the increases in species richness were caused by only a few individuals that only occur ephemerally at restored sites, there would be a marked decrease in evenness, which was not observed until 28 weeks after restoration.

The cause of the unusually high diversity at restored living shorelines may, in part, be due to the methods of shoreline stabilization. During restoration 2-gallon mesh bags filled with disarticulated oyster shell are placed in the intertidal to attenuate wave energy, allowing newly planted marsh grass and mangroves to survive in early restoration stages. Though control shorelines are considered pristine, they do not have these oyster bags providing additional habitat as restored sites do. As mangroves begin to overgrow these bags and they begin to become buried by constant wave action, diversity may plateau and reflect similar levels to control sites. Including additional data collected after a greater time following restoration may corroborate this hypothesis, again emphasizing the need for long-term monitoring to fully quantify the success of restoration.

Hypotheses regarding community composition on restored oyster reefs were not, for the most part, corroborated by the findings of this study. One exception being the significant difference between live reef communities and dead reef communities. Significant differences in community composition were detected between live and restored reefs at the 1-week, 4-week, and 40-week sampling times but otherwise were not observed across treatment-times between the two site types. Additionally, live and dead reefs harbored significantly different communities only at the 40-week sampling period within treatment-times. Similar patterns were observed at living shorelines. Though restored and control sites produced significantly different communities as a whole, restored and control treatments yielded significantly different communities only at the 1-week and 4-week treatment-times. Once again, treatment-times may be suffering from a lack of statistical power due to small sample sizes. The inclusion of year 2 data as replicates may be necessary detect fine-scale changes in macroinvertebrate communities post-restoration.

Environmental conditions can also have significant effects on diversity and species composition (MacArthur, 1964; Fuhrman et al., 2008; Bouskill et al., 2012). Low temperatures during the 28- and 40-week sampling periods, which were conducted during January and April 2018, may have influenced species diversity and composition across treatments. Interestingly, high-quality habitats seem to be more resistant to seasonal fluctuations in diversity than lower-quality habitats, such as dead reefs and restored reefs. For example, dead oyster reefs experienced the largest decrease in species richness of all treatments while live reefs showed little to no change during the 28 and 40-week time periods. Restored reefs showed a lag in species richness declines, which may have been caused by intermediate levels of oyster densities. Living shorelines showed similar temporal trends in diversity with minima at 28 and 40 weeks after restoration. Restored sites harbored higher species diversity than control sites at these times but still experienced large declines. The restoration of living shorelines does not involve recruitment of intertidal foundation species as oyster reefs do but rather places emphasis on the stabilization of local sediments and attenuation of wave action. This discrepancy in restoration strategies may explain the differences between the live and control sites at oyster reefs and living shorelines. However, verification of these hypotheses is beyond the scope of this thesis without incorporation of models quantifying the effect of environmental parameters on diversity and abundance of macroinvertebrates.

The findings of this study have elucidated how specific ecosystem components can change after habitat restoration. Though not statistically detectable, species richness and Shannon diversity did increase from background levels on dead oyster reefs to background levels on live reefs suggesting that habitat restoration can facilitate the recovery of some aspects of natural communities in relatively short periods of time (i.e. 1 year). However, similarities in communities between treatment-times within the 1-year study period reemphasizes the importance of long-term monitoring in assessing restoration success. Living shoreline sites did not display differences between treatments before or after restoration. For this reason, biotic community-based metrics may not be appropriate for assessing the efficacy of restoration of living shoreline habitats, whereas abiotic metrics (i.e. sediment dynamics and wave energy dispersion) may yield more useful data. Based on these results, more funding and effort should be dedicated to monitoring restoration projects in the future to truly define what can be considered a successful restoration and what quantifiable metrics are ideal for assessing site-specific restoration goals on a case-by-case basis. Additionally, some changes in communities may require thorough sampling regimes to detect statistically significant differences. Therefore, ensuring adequate sampling throughout restoration monitoring is essential.

Efficient and successful habitat restoration efforts will be crucial to the survival of our increasingly degraded biosphere. Further analyses will be conducted to quantify the relationship between habitat metrics, such as oyster densities and rugosities as well as sedimentation dynamics, and species diversity over time. Models that integrate unique components of the ecosystem will help researchers and management agencies effectively assess and monitor the success of future restorations. These data will be used as part of a collaborative interdisciplinary Coupled Natural-Human Systems grant from the National Science Foundation. Studies like these are necessary to ensure holistic evaluations of recovery in ecosystem services and vital ecological processes. By taking into account multiple ecological components, we can improve the efficacy by which we restore and monitor degraded habitats. Thereby, augmenting the efficiency by which we can reverse habitat losses in an increasingly exploited biosphere.

APPENDIX: SIMPER TABLES

Table 5. Average abundances and contributions of species by time at live oyster reefs.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Live					
Before Restoration					
Petrolisthes armatus	0.19	1.77	0.21	36.88	36.88
Alpheus heterochaelis	0.28	1.27	0.24	26.39	63.27
Panopeus herbstii	0.17	1.19	0.19	24.75	88.02
Callinectes ornatus	0.1	0.23	0.13	4.74	92.76
1 Week					
Petrolisthes armatus	0.37	3.57	0.34	47.22	47.22
Eurypanopeus depressus	0.26	2.05	0.26	27.16	74.38
Panopeus herbstii	0.23	1.01	0.19	13.33	87.71
Periclimenaeus spp	0.23	0.51	0.12	6.72	94.43
2 Weeks					
Petrolisthes armatus	0.51	4.27	0.45	43.99	43.99
Panopeus herbstii	0.33	2.07	0.38	21.34	65.33
Rhithropanopeus harrisii	0.28	1.46	0.3	15.06	80.39
Periclimenaeus spp	0.24	0.68	0.16	7.03	87.42
Eurypanopeus depressus	0.14	0.63	0.22	6.5	93.92
4 Weeks					
Petrolisthes armatus	0.85	7.49	0.54	53.23	53.23
Panopeus herbstii	0.39	2.24	0.38	15.91	69.15
Rhithropanopeus harrisii	0.36	1.78	0.33	12.62	81.77
Eurypanopeus depressus	0.29	1.72	0.34	12.23	94.01
28 Weeks					
Petrolisthes armatus	1.48	15.15	0.72	69.04	69.04
Eurypanopeus depressus	0.59	4.99	0.52	22.72	91.77
40 Weeks					
Petrolisthes armatus	1.82	22.12	0.99	76.06	76.06
Eurypanopeus depressus	0.43	3.58	0.5	12.31	88.38
Panopeus herbstii	0.35	1.67	0.31	5.75	94.12
48 Weeks					
Petrolisthes armatus	1.6	16.26	0.88	67.49	67.49
Eurypanopeus depressus	0.52	2.61	0.43	10.84	78.33
Panopeus simpsoni	0.49	2.28	0.43	9.48	87.81
Rhithropanopeus harrisii	0.36	1.12	0.28	4.65	92.46

Table 6. Average abundances and contributions of species by time at restored oyster reefs.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Restored					
Before Restoration					
Petrolisthes armatus	0.4	2.35	0.22	72.93	72.93
Panopeus herbstii	0.11	0.46	0.13	14.26	87.2
Eurypanopeus depressus	0.09	0.15	0.09	4.73	91.92
1 Week					
Panopeus herbstii	0.16	1.29	0.24	40.38	40.38
Petrolisthes armatus	0.16	1.29	0.24	40.38	80.75
Panopeus bermudensis	0.08	0.27	0.13	8.46	89.22
Periclimenaeus spp	0.08	0.27	0.13	8.46	97.68
2 Weeks					
Petrolisthes armatus	0.19	0.97	0.23	28.98	28.98
Rhithropanopeus harrisii	0.13	0.78	0.23	23.4	52.38
Periclimenaeus spp	0.16	0.76	0.17	22.8	75.17
Panopeus herbstii	0.09	0.26	0.12	7.84	83.01
Panopeus simpsoni	0.04	0.26	0.07	7.65	90.66
4 Weeks					
Panopeus herbstii	0.19	1.77	0.26	44.02	44.02
Rhithropanopeus harrisii	0.33	1.3	0.19	32.35	76.37
Petrolisthes armatus	0.18	0.46	0.14	11.53	87.9
Clibinarius vittatus	0.06	0.22	0.11	5.56	93.45
28 Weeks					
Petrolisthes armatus	1.81	17.17	0.75	80.25	80.25
Rhithropanopeus harrisii	0.44	1.31	0.3	6.12	86.37
Eurypanopeus depressus	0.35	1.24	0.26	5.77	92.14
40 Weeks					
Petrolisthes armatus	0.9	7.16	0.44	72.15	72.15
Panopeus herbstii	0.24	1.53	0.29	15.46	87.6
Eurypanopeus depressus	0.17	0.72	0.17	7.27	94.87
48 Weeks					
Petrolisthes armatus	1.78	19.2	0.86	78.84	78.84
Panopeus simpsoni	0.36	1.45	0.32	5.95	84.79
Eurypanopeus depressus	0.24	1.09	0.25	4.48	89.27
Panopeus herbstii	0.21	0.87	0.24	3.56	92.83

Table 7. Average abundances and contributions of species by time at dead oyster reefs.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Dead					
Before Restoration					
Panopeus herbsii	0.29	1.97	0.27	39.22	39.22
Petrolisthes armatus	0.26	1.42	0.19	28.42	67.64
Rhithropanopeus harrisii	0.18	1.19	0.23	23.74	91.38
1 Week					
Petrolisthes armatus	0.38	3.68	0.31	47.78	47.78
Panopeus herbstii	0.29	3.68	0.33	47.74	95.52
2 Weeks					
Petrolisthes armatus	0.4	2.38	0.3	55.02	55.02
Panopeus herbstii	0.2	1.03	0.21	23.94	78.97
Eurypanopeus depressus	0.14	0.6	0.22	13.8	92.77
4 Weeks					
Petrolisthes armatus	0.69	3.51	0.3	57.74	57.74
Panopeus herbstii	0.28	1.49	0.3	24.43	82.17
Rhithropanopeus harrisii	0.31	0.6	0.18	9.93	92.11
28 Weeks					
Petrolisthes armatus	1.46	18.1	0.66	89.26	89.26
Eurypanopeus depressus	0.24	1.35	0.32	6.65	95.9
40 Weeks					
Petrolisthes armatus	1.03	8.99	0.47	75.27	75.27
Eurypanopeus depressus	0.37	2.03	0.37	17.02	92.29
48 Weeks					
Petrolisthes armatus	1.36	11.69	0.65	70.12	70.12
Panopeus simpsoni	0.57	3.69	0.47	22.11	92.23

Table 8. Average abundances and contributions of species by time at control living shorelines.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Control					
Before Restoration					
Alpheus heterochaelis	1.01	8.45	0.84	40.35	40.35
Petrolisthes armatus	1.44	4.66	0.44	22.27	62.62
Rhithropanopeus harrisii	0.64	3.06	0.53	14.63	77.26
Periclimenaeus spp	0.49	1.22	0.32	5.81	83.07
Leander spp	0.63	1.18	0.2	5.65	88.72
Eurypanopeus depressus	0.48	1.16	0.32	5.54	94.26
1 Week					
Petrolisthes armatus	0.99	6.24	0.48	66.63	66.63
Rhithropanopeus harrisii	0.26	0.91	0.29	9.76	76.38
Palaemon spp	0.12	0.69	0.14	7.41	83.8
Eurypanopeus depressus	0.22	0.66	0.22	7.08	90.88
2 Weeks					
Petrolisthes armatus	1.14	8.43	0.89	26.46	26.46
Alpheus heterochaelis	0.85	5.79	0.73	18.18	44.64
Eurypanopeus depressus	0.83	5.26	0.55	16.53	61.17
Rhithropanopeus harrisii	0.8	4.6	0.7	14.45	75.62
Leander spp	1.01	3.17	0.33	9.95	85.57
Palaemonetes spp	0.65	2.75	0.32	8.65	94.21
4 Weeks					
Rhithropanopeus harrisii	1.07	6.78	0.69	32.01	32.01
Alpheus heterochaelis	1.03	6.18	0.73	29.15	61.15
Petrolisthes armatus	0.7	3.94	0.52	18.57	79.72
Eurypanopeus depressus	0.55	2.79	0.53	13.18	92.9
28 Weeks					
Petrolisthes armatus	1.14	7.87	0.62	51.86	51.86
Eurypanopeus depressus	0.88	5.39	0.54	35.55	87.41
Rhithropanopeus harrisii	0.2	0.62	0.22	4.09	91.5
40 Weeks					
Eurypanopeus depressus	0.64	11.53	0.52	75.36	75.36
Petrolisthes armatus	0.59	2.79	0.29	18.25	93.61
48 Weeks					
Petrolisthes armatus	1.56	10.63	0.76	45.94	45.94
Alpheus heterochaelis	0.68	4.26	0.62	18.42	64.36
Callinectes sapidus	0.47	2.77	0.55	11.96	76.32
Palaemon spp	0.59	2.71	0.39	11.71	88.04
Eurypanopeus depressus	0.46	1.44	0.4	6.2	94.24

Table 9. Average abundances and contributions of species by time at restored living shorelines.

Species	Av. Abun.	Av. Sim.	Sim./SD	Contr. %	Cum. %
Restored					
Before Restoration					
Petrolisthes armatus	1.11	11.25	1.03	40.41	40.41
Alpheus heterochaelis	0.83	6.77	0.76	24.33	64.74
Leander spp	0.9	2.72	0.33	9.75	74.49
Periclimenaeus spp	0.69	2.59	0.43	9.32	83.81
Palaemon spp	0.65	2.1	0.3	7.53	91.34
1 Week					
Petrolisthes armatus	1.53	20.11	0.99	74.61	74.61
Leander spp	0.51	2.62	0.29	9.73	84.33
Eurypanopeus depressus	0.34	1.49	0.31	5.51	89.85
Palaemonetes spp	0.44	1.07	0.18	3.98	93.83
2 Weeks					
Petrolisthes armatus	1.61	9.69	0.87	39.86	39.86
Eurypanopeus depressus	1.02	5.56	0.76	22.89	62.75
Rhithropanopeus harrisii	0.9	3.8	0.56	15.63	78.38
Leander spp	0.95	2.2	0.35	9.03	87.42
Alpheus heterochaelis	0.42	0.87	0.33	3.58	91
4 Weeks					
Petrolisthes armatus	1.45	13.68	1.24	33.49	33.49
Eurypanopeus depressus	1.27	11.32	1.08	27.72	61.2
Rhithropanopeus harrisii	1.25	8.08	0.81	19.78	80.99
Alpheus heterochaelis	0.67	4.12	0.69	10.09	91.07
28 Weeks					
Petrolisthes armatus	1.31	14.07	0.73	63.22	63.22
Eurypanopeus depressus	0.8	6.05	0.62	27.21	90.43
40 Weeks					
Eurypanopeus depressus	0.62	18.4	0.77	69.68	69.68
Petrolisthes armatus	0.8	7.91	0.46	29.96	99.64
48 Weeks					
Petrolisthes armatus	2.11	16.44	1.02	44.6	44.6
Eurypanopeus depressus	1.16	8.25	0.94	22.38	66.98
Palaemon spp	0.9	6.36	0.65	17.25	84.22
Callinectes sapidus	0.54	3.3	0.61	8.96	93.18

REFERENCES

Barber, A., Walters, L., and Birch, A. (2006). Potential for Restoring Biodiversity of Macroflora and Macrofauna On Oyster Reefs in Mosquito Lagoon, Florida. *Florida Scientist*, 73(1), 47-62.

Bernhardt, E. S., Sudduth, E. B., Palmer, M. A., Allan, J. D., Meyer, J. L., Alexander, G., and Rumps, J. (2007). Restoring rivers one reach at a time: results from a survey of US river restoration practitioners. *Restoration Ecology*, 15(3), 482-493.

Boudreaux, M.L., Stiner, J.L., Walters, L.J. (2006) Biodiversity of Sessile And Motile Macrofauna On Intertidal Oyster Reefs in Mosquito Lagoon, Florida. *Journal of Shellfish Research*, 25(3), 1079-1089.

Bouskill, N. J., Eveillard, D., Chien, D., Jayakumar, A., and Ward, B. B. (2012). Environmental factors determining ammonia-oxidizing organism distribution and diversity in marine environments. *Environmental Microbiology*, 14(3), 714-729.

Bray, J. R., and Curtis, J. T. (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecological monographs*, 27(4), 325-349.

Breitburg, D. L. (1999). Are Three-Dimensional Structure And Healthy Oyster Populations the Keys to An Ecologically Interesting And Important Fish Community? In Luckenbach, M. W., R. Mann, and J. A. Wesson, editors. Oyster reef habitat restoration: a synopsis and synthesis of approaches. *Virginia Institute of Marine Science Press*. Gloucester Point, Virginia. pp. 239–250.

Britto, V.O. and Bugoni, L. Hydrobiologia (2015). The Contrasting Feeding Ecology of Great Egrets and Roseate Spoonbills In Limnetic And Estuarine Colonies. *Hydrobiologia*, 744(1), 187-210.

Brooks, T. M., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A., Rylands, A. B., Konstant, W. R., and Hilton-Taylor, C. (2002). Habitat Loss and Extinction in The Hotspots of Biodiversity. *Conservation biology*, *16*(4), 909-923.

Cabaço, S., Santos, R., and Duarte, C. M. (2008). The impact of sediment burial and erosion on seagrasses: a review. *Estuarine, Coastal and Shelf Science*, 79(3), 354-366.

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Austral ecology*, 18(1), 117-143.

Coasts, Oceans, Ports, and Rivers Institute (COPRI). 2014. Living shorelines database. http://mycopri.org/node/1467

Coen, L. D., Knott, D. M., Wenner, E. L., Hadly, N. H., Ringwold, A. H., and Bobo, M. Y. (1996). Intertidal Oyster Reef Studies In South Carolina: Design, Sampling and Experimental Focus for Evaluating Habitat Value and Function. In: M.W. Luckenbach, R. Mann and J. A. Wesson, editors. Oyster reef habitat restoration: a synopsis and synthesis of approaches. Gloucester Point, Virginia. *Virginia Institute of Marine Science Press*. pp. 239–250.

Cole, K.M., Supan, J., Ramirez, A. and Johnson, C.N. (2015). Suspension of Oysters Reduces The Populations Of *Vibrio parahaemolyticus* And *Vibrio vulnificus*. *Letters In Applied Microbiology*, 61, 209–213.

Connolly, L.M., and Colwell, M.A. (2005). Comparative Use Of Longline Oyster beds And Adjacent Tidal Flats By Waterbirds. *Bird Conservation International*, 15(3): 237-255.

Crabtree, R. E. & J. M. Dean. (1982). The Structure of Two South Carolina Estuarine Tide Pool Fish Assemblages. *Estuaries,* 5, 2–9.

Crooks, S., and Turner, R. K. (1999). Integrated coastal management: sustaining estuarine natural resources. Pages 241 – 289 in D. B. Nedwell , and D. G. Raffaelli. Advances in ecological research, vol. 29. *Academic Press*, San Diego, California, USA.

Davis, J. L., Takacs, R. L., and Schnabel, R. (2006). Evaluating ecological impacts of living shorelines and shoreline habitat elements: an example from the upper western Chesapeake Bay. *Management, policy, science, and engineering of nonstructural erosion control in the Chesapeake Bay*, 55.

Donnelly, M., Shaffer, M., Connor, S., Sacks, P., and Walters, L. (2017). Using Mangroves To Stabilize Coastal Historic Sites: Deployment Success Versus Natural Recruitment. *Hydrobiologia*, 803(1): 389-401.

Dorenbosch, M., Grol, M.G.G., Nagelkerken, I., and van der Velde, G. (2006). Different Surrounding Landscapes May Result in Different Fish Assemblages in East African Seagrass Beds. *Hydrobiologia*, 563, 45-60.

Fuhrman, J. A., Steele, J. A., Hewson, I., Schwalbach, M. S., Brown, M. V., Green, J. L., and Brown, J. H. (2008). A latitudinal diversity gradient in planktonic marine bacteria. *Proceedings of the National Academy of Sciences*, 105(22), 7774-7778.

Gain, I.E., Brewton, R.A., Robillard, M.M.R., Johnson, K.D., Smee, D.L., and Stunz, G.W. (2017). Macrofauna Using Intertidal Oyster Reef Varies in Relation to Position Within the Estuarine Habitat Mosaic. *Marine Biology*, 164, 8.

Garvis, S.K., Sacks, P.E., and Walters, L.J. (2015). Formation, Movement, and Restoration of Dead Intertidal Oyster Reefs in Canaveral National Seashore and Mosquito Lagoon, Florida. *Journal of Shellfish Biology*, 34(2), 251-258.

Gaston, K. J., Blackburn, T. M., & Goldewijk, K. K. (2003). Habitat conversion and global avian biodiversity loss. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, *270*(1521), 1293-1300.

Geraldi, N. R., Powers, S. P., Heck, K. L., and Cebrian, J. (2009). Can habitat restoration be redundant? Response of mobile fishes and crustaceans to oyster reef restoration in marsh tidal creeks. *Marine Ecology Progress Series*, 389, 171-180.

Gilmore, R. G. 1995. Environmental and biogeographic factors influencing ichthyofaunal diversity: Indian River Lagoon. *Bulletin of Marine Science,* 57, 153–170.

Gittman, R. K., Peterson, C. H., Currin, C. A., Joel Fodrie, F., Piehler, M. F., and Bruno, J. F. (2016). Living shorelines can enhance the nursery role of threatened estuarine habitats. *Ecological Applications*, *26*(1), 249-263.

Goldewijk, K. K. (2001). Estimating global land use change over the past 300 years: the HYDE database. *Global biogeochemical cycles*, *15*(2), 417-433.

Grimbacher, P. S., & Catterall, C. P. (2007). How much do site age, habitat structure and spatial Grimbacher, P. S., & Catterall, C. P. (2007). How much do site age, habitat structure and spatial isolation influence the restoration of rainforest beetle species assemblages?. *Biological Conservation*, *135*(1), 107-118.

Hanski, I. (2011). Habitat loss, the dynamics of biodiversity, and a perspective on conservation. *Ambio*, *40*(3), 248-255.

Hellmann, R. (2013). Canaveral National Seashore: Archeological Overview and Assessment. Southeast Archeological Center, National Park Service, Tallahassee, FL.

Hobbs, R. J. and Norton, D. A. (1996). Towards a conceptual framework for restoration ecology. *Restoration ecology*, *4*(2), 93-110.

Hoekstra, J. M., Boucher, T. M., Ricketts, T. H., & Roberts, C. (2005). Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology letters*, *8*(1), 23-29.

Huxel, G. R., & Hastings, A. (1999). Habitat loss, fragmentation, and restoration. *Restoration ecology*, *7*(3), 309-315.

Jones, C. G., Lawton, J. H., Shachak, M. (1994). Organisms as Ecosystem Engineers. *Oikos,* 69:373-386.

Luckenbach, M. W., Coen, L. D., Ross Jr, P. G., & Stephen, J. A. (2005). Oyster reef habitat restoration: relationships between oyster abundance and community development based on two studies in Virginia and South Carolina. *Journal of Coastal Research*, 64-78.

Lucrenzi, S., Schlacher, T. A., and Robinson, W. (2010). Can storms and shore armouring exert additive effects on sandy- beach habitats and biota? *Marine and Freshwater Research,* 61: 951.

MacArthur, R. H. (1964). Environmental factors affecting bird species diversity. *The American Naturalist*, *98*(903), 387-397.

Marbà, N., and Duarte, C. M. (1994). Growth response of the seagrass Cymodocea nodosa to experimental burial and erosion. *Marine Ecology Progress Series*, 307-311.

McDonald, J. (1982). Divergent life history patterns in the co-occurring intertidal crabs Panopeus herbstii and Eurypanopeus depressus (Crustacea: Brachyura: Xanthidae). *Marine Ecology Progress Series*, 173-180.

Menzel, W.R., and Hopkins, S.H. (1956). Crabs as predators of oysters in Louisiana. *Proceedings of the National Shellfisheries Association*, 46, 177–184.

Meyer, D. L. (1994). Habitat partitioning between the xanthid crabs *Panopeus herbstii* and *Eurypanopeus depressus* on intertidal oyster reefs (*Crassostrea virginica*) in southeastern North Carolina. *Estuaries*, 17, 674–679.

Miller, S. W., Budy, P., & Schmidt, J. C. (2010). Quantifying macroinvertebrate responses to in-stream habitat restoration: applications of meta-analysis to river restoration. *Restoration Ecology*, *18*(1), 8-19.

Nagendra, H. (2002). Opposite trends in response for the Shannon and Simpson indices of landscape diversity. *Applied Geography*, *22*(2), 175-186.

Paperno, R., Millie, K.J., and Kadison, E. (2001). Patterns In Species Composition if Fish and Selected Invertebrate Assemblages in Estuarine Subregions Near Ponce De Leon Inlet, Florida. *Estuarine, Coastal and Shelf Science*, 52, 117-130.

Piazza, B. P., P. D. Banks , and M. K. La Peyre . 2005. The potential for created oyster shell reefs as a sustainable shoreline protection strategy in Louisiana. *Restoration Ecology,* 13, 499 – 506.

Plunket, J. and La Peyre, M.K. (2005). Oyster Beds as Fish and Macroinvertebrate Habitat In Barataria Bay. *Bulletin of Marine Science*, 77(1), 155-164.

Rees, G. N., Baldwin, D. S., Watson, G. O., Perryman, S., & Nielsen, D. L. (2004). Ordination and significance testing of microbial community composition derived from terminal restriction fragment length polymorphisms: application of multivariate statistics. *Antonie Van Leeuwenhoek*, *86*(4): 339-347.

Rueda, M., and Defeo, O. (2003). Spatial structure of fish assemblages in a tropical estuarine lagoon: combining multivariate and geostatistical techniques. *Journal of Experimental Marine Biology and Ecology*, 296(1), 93-112.

Ruiz, J.M., Romero, J., (2003). Effects of disturbances caused by coastal constructions on spatial structure, growth dynamics and photosynthesis of the seagrass *Posidonia oceanica*. *Marine Pollution Bulletin,* 46, 1523–1533.

Seitz, R., Lipcius, R., Olmstead, N., Seebo, M., and Lambert, D. (2006). Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. *Marine Ecology Progress Series*, 326, 11 – 27.

Shervette, V.R. and Gelwick, F. (2008). Seasonal and Spatial Variation in Fish and Macroinvertebrate Communities of Oyster and Adjacent Habitats in a Mississippi Estuary. *Estuary and Coasts*, 31, 584-596.

Shannon, C. E. and Wiener, W. (1963). The mathematical theory of communication. University *Illinois Press*, Urbana, 360 pp.

Short, F., Carruthers, T., Dennison, and W., Waycott, M. (2007). Global Seagrass Distribution And Diversity: A Bioregional Model. *Journal of Experimental Marine Biology and Ecology*, 350(1–2), 3-20.

Simpson, E.H. (1949). Measurement of diversity. *Nature*, 163(4148), 688.

Smithsonian Institution. (2006). Indian River Lagoon Species Inventory Report. Website: http://www.sms.si.edu/irlspec/index.htm.

Swain, H. M. (1995). Reconciling rarity and representation: a review of listed species in the Indian River Lagoon. *Bulletin of Marine Science*, *57*(1), 252-266.

Swann, L. (2008). The use of living shorelines to mitigate the effects of storm events on Dauphin Island, Alabama, USA. In *American Fisheries Society Symposium*, 64(11).

Tolley, S. G. and A. K. Volety. (2005). The role of oysters in habitat use of oyster reefs by resident fishes and decapod crustaceans. *Journal of Shellfish Research*, 24, 1007–1012.

Tremain, D. M., & Adams, D. H. (1995). Seasonal variations in species diversity, abundance, and composition of fish communities in the northern Indian River Lagoon, Florida. *Bulletin of Marine Science*, *57*(1), 171-192.

Van Katwijk, M. M., Bos, A. R., De Jonge, V. N., Hanssen, L. S. A. M., Hermus, D. C. R., & De Jong, D. J. (2009). Guidelines for seagrass restoration: importance of habitat selection and donor population, spreading of risks, and ecosystem engineering effects. *Marine pollution bulletin*, 58(2), 179-188.

Yeager, L.A. and Layman, C.A. (2011). Energy Flow to Two Abundant Consumers in A Subtropical Oyster Reef Food Web. *Aquatic Ecology*, 45, 267-277.